A peer-reviewed open-access journal
NeoBiota
Advancing research on alien species and biological invasions

NeoBiota 94: 311–331 (2024)
DOI: 10.3897/neobiota.94.123363

Research Article

Uprooting is a promising tool to control invasive giant ragweed and recover native diversity

Chaeho Byun[1], Kripal Singh[1], Sun Hee Hong[2], Jangho Lee[1], Tae Kyung Yoon[3], Hojeong Kang[4]

1 *Department of Biological Sciences, Andong National University, Andong 36729, Republic of Korea*
2 *School of Plant Science and Landscape Architecture, Hankyong National University, Anseong-si 17579, Gyeonggi-do, Republic of Korea*
3 *Department of Forest Science, Sangji University, Wonju 26339, Republic of Korea*
4 *School of Civil and Environmental Engineering, Yonsei University, Seoul 03722, Republic of Korea*

Corresponding author: Chaeho Byun (chaehobyun@anu.ac.kr, chaeho.byun@mail.mcgill.ca)

Abstract

Ambrosia trifida (giant ragweed) is an invasive species that causes habitat destruction and competitively excludes native plants in many parts of Europe and Asia. In this study, we evaluated the effects of selective cutting and uprooting on *A. trifida* and native plant diversity, as well as the effects of sowing the seeds of native annual, perennial and woody species after eradication. We hypothesised that: (i) selective uprooting will be more effective than cutting in controlling invasion by *A. trifida* because fewer propagules would be left behind, with no increase in the number of existing invasive propagules and (ii) sowing native seeds will increase invasion resistance and accelerate the recovery of native plant diversity. The eradication methods were applied in July 2022, seeds were sown in March 2023 and the response variables (i.e. importance values (%) of *A. trifida* and diversity index (H') of native species) were measured in September 2023. The importance values of *A. trifida* were lowest and diversity index of native species was highest in the uprooting treatment, supporting the first hypothesis. Sowing native seeds following invasion removal did not exert significant additional suppressive effects on invasion or increase native species diversity. These results reveal that selective uprooting is a promising tool to control *A. trifida* and to support the recovery of native diversity, while sowing native seeds does not improve the quality of restoration.

Key words: *Ambrosia trifida*, ecological restoration, eradication methods, diversity index, hand-pulling, native diversity, invasive plant management, selective cutting, selective uprooting

OPEN ACCESS

Academic editor: Graeme Bourdôt
Received: 18 March 2024
Accepted: 1 August 2024
Published: 23 August 2024

Citation: Byun C, Singh K, Hong SH, Lee J, Yoon TK, Kang H (2024) Uprooting is a promising tool to control invasive giant ragweed and recover native diversity. NeoBiota 94: 311–331. https://doi.org/10.3897/neobiota.94.123363

Copyright: © Chaeho Byun et al.
This is an open access article distributed under terms of the Creative Commons Attribution License (Attribution 4.0 International – CC BY 4.0).

Introduction

Biological invasion is a major factor contributing to global ecological and socioeconomic instability (Simberloff et al. 2013; Roy et al. 2023). Invasive alien plants cause biodiversity loss through the competitive exclusion of native species, ultimately leading to the loss of valuable endemic species (Vilà et al. 2011; Jaureguiberry et al. 2022; Adhikari et al. 2023). Invasive plants often disrupt ecosystem processes (Pearson et al. 2018), such as nutrient and water cycling (Vilà et al. 2011) and alter the soil composition (Castro-Díez et al. 2019). Several plant species in introduced areas produce allergens (Shackleton et al. 2016) and cause severe health issues. Shifts in biological integrity and natural ecosystem processes due to invasive plants affect the supply of ecosystem services, livelihoods and

human well-being (Shackleton et al. 2018). Therefore, managing plant invasion is crucial for conserving biodiversity and sustaining natural ecosystems and human health. The management of invasive species includes the early identification of new invasive species, prevention of spread, eradication of existing invasive species and subsequent restoration initiatives (Pyšek and Richardson 2010).

Ambrosia trifida L., or giant ragweed, is an annual herb native to North America (Laforest et al. 2024). It is adapted to a wide range of habitats, including agricultural fields, roadsides and disturbed areas, contributing to its invasive success in Europe and East Asia (Abul-Fatih and Bazzaz 1979; Soltani et al. 2011; Chauvel et al. 2021). Its rapid growth, high seed production and tolerance to a variety of ecosystem disturbances and environmental conditions allow it to outcompete native plants and associated fauna (Regnier et al. 2016; Dong et al. 2020a; Chauvel et al. 2021; Wang et al. 2022; Xian et al. 2023; Chen et al. 2024). The plant was unintentionally introduced into the Republic of Korea in 1970, probably as a contaminant of agricultural equipment and/or crop seed. It has been considered an invasive species since 1999 (Park et al. 2012); initially, it was discovered near the demilitarised zone, in the central region of Korea (Lee et al. 2010). After it spread throughout the country, it was registered as an ecosystem-disturbing species (Lee et al. 2010; Yin et al. 2010; Kim 2017; Montagnani et al. 2017; Li et al. 2022). *Ambrosia trifida* has high reproductive ability and is difficult to remove once established. It grows quickly, attains heights of up to 5 m and forms high densities, greatly reducing the growth of native species and economic crops in cultivated areas. Pollen from flowers of *A. trifida* can also cause allergies (Ling et al. 2022). Therefore, the eradication of *A. trifida* and restoration of native diversity is of paramount socioeconomic and ecological importance.

A number of mechanical (physical), chemical and biological eradication methods for invasive plant species have been developed (Kettenring and Adams 2011; Weidlich et al. 2020), with variable outcomes in terms of invasion control (Pearson et al. 2016; Chenot et al. 2018; Courkamp et al. 2022) and recovery of native plant communities (Boxriker et al. 2022; Farmilo and Moxham 2023). The mechanical approach involves the physical removal of invasive species through cutting, mowing, uprooting, burning, mechanical harvest and other similar approaches. Except in cases where uprooting is applied, re-sprouting and habitat disturbance facilitate re-invasion and hinder the recovery of native communities (Schooler et al. 2010; Byun et al. 2020a; Nagy et al. 2022). Chemical application is the most frequently used method (Kettenring and Adams 2011) to eradicate specific plant species before and/or after germination. However, the application of herbicides, such as glyphosate, indaziflam, 2,4-2,4-dichlorophenoxyacetic acid and picloram, deters native vegetation and fauna (Carlson and Gorchov 2004; Robichaud and Rooney 2021; Donaldson and Germino 2022) and many nations banned their use to control weeds and invasive species (Peng et al. 2020; Pergl et al. 2020). Although the introduction of natural enemies, such as insects and pathogens (bacteria, fungi, viruses etc.), can hinder the germination, growth and spread of invasive plants (Clewley et al. 2012), this approach is logistically complex and not well investigated (David and Lake 2023; Shen et al. 2023). The mechanical method in which invasive species are uprooted is considered a new tool to control invasion (Kollmann et al. 2011) and to have multiple advantages compared with conventional methods, such as cutting because it does not leave any reproductive or regenerative parts of plants behind. This is important because this species can regrow from its roots or stem fragments after cutting or disturbance, which contributes to its resilience and ability to spread

rapidly, making it a challenging invasive species to manage. However, selective uprooting can be labour-intensive and time-consuming and its efficiency is unclear, in part owing to the small number of relevant studies (Pickart et al. 1998a; Pickart et al. 1998b; Ussery and Krannitz 1998). Recent empirical studies, systematic reviews and meta-analyses clearly indicate that the use of control methods alone does not facilitate the recovery of native communities, suggesting that it is imperative to consider active restoration initiatives (Kettenring and Adams 2011; Adams et al. 2020; Singh and Byun 2023). For example, sowing native seeds can contribute to the recovery of native communities (Kettenring and Adams 2011).

Extensive research suggests that sowing native seeds after the removal of invasive species can establish native vegetation cover and reduce the probability that invasive species regrow and establish as a result of niche pre-emption and resource utilisation (Levine et al. 2004; Enloe et al. 2005; Sheley et al. 2006). Therefore, native seed addition following invasion control is essential for restoring native diversity (O'Donnell et al. 2016; Johnson et al. 2018). The presence of functionally similar and dissimilar species can provide clues about the potential of various native species to protect against invasion (Sheley and James 2010; Byun et al. 2013). However, Yannelli and colleagues reported that seed density (propagule pressure in ecological terms) is more effective than trait similarity with respect to invasion suppression (Yannelli et al. 2020). Therefore, it is critical to determine if sowing native seeds suppresses invasion and facilitates the recovery of native diversity, as well as the effects of species of different functional groups.

Ambrosia trifida is a noxious weed and its control has been a challenging task at introduced sites and in its native range. The suppression of *A. trifida* in agricultural fields by the application of various herbicides, growing cover crops and diversifying cropping systems is rarely successful (Kouame et al. 2023; Silva et al. 2023; Werle et al. 2023). Similarly, in natural ecosystems, such as grasslands, the impact of mechanical eradication methods on the spread of *A. trifida* varies and the approach is generally either ineffective or only slightly effective (Byun and Lee 2018; Park et al. 2020; Byun et al. 2020b; Byun 2023). Intense mowing management of *A. trifida* followed by sowing native seed suppressed invasion up to 77%; however, seeding did not amplify suppression or facilitate native recovery (Byun et al. 2020b). The competitive ability of *A. trifida* was not suppressed, even when grown with eight native species from three functional groups (Byun and Lee 2018). However, the effectiveness of selectively targeting only invasive species (*A. trifida* in this study) by cutting (removing aboveground plant parts) or uprooting (pulling out whole plant) has not been tested. In addition, the impact of sowing native seeds following the selective eradication of *A. trifida* on native diversity is not known.

This study, therefore, aimed to investigate the effectiveness of mechanical control, including selective uprooting and selective cutting, on the dominance of *A. trifida*. Although the eradication of invasive species can suppress their dominance to some extent, the recovery of native diversity requires additional assistance (e.g. through sowing seeds following invasive plant removal) (Enloe et al. 2005; Sheley et al. 2006; Kettenring and Adams 2011; Adams et al. 2020; Singh and Byun 2023). Sowing seeds of various functional groups can further protect against invasion through trait similarity (annuals), structural asymmetry (perennials) and diversity (three species). Therefore, we sowed seeds of nine native species from three functional groups (i.e. annual, perennial and woody species) after invasion removal to evaluate whether there are additional benefits in the suppression of

A. trifida invasion. We hypothesised that selective uprooting will be more effective than cutting in controlling invasion by *A. trifida*, because uprooting leaves fewer propagules behind and does not add to the number of existing propagules. We also hypothesised that sowing native seeds will protect against invasion and suppress *A. trifida* through limiting similarity with annuals and increasing diversity and will accelerate the recovery of native diversity.

Methods

Study site

Experimental plots were installed in July 2022 at two sites in Busan, Republic of Korea separated by 18 km. Site #1 was located at 1200-5 Daejeo 2-dong, Gangseo-gu (35°11'46"N, 128°58'02"E) and site #2 was located at 1211 Hadan-dong (35°05'29"N, 128°56'40"E) (Fig. 1). The sites represent a typical riparian habitat that was originally dominated by *A. trifida* (> 90% cover).

Removal of invasive *A. trifida* and sowing native seeds

We applied a split-plot design with main plots consisting of different eradication methods and subplots consisting of different seed mixtures for sowing. Based on the cover and distribution of *A. trifida*, two blocks at site 1 and four blocks at site 2 were established. Thus, a total of six blocks were prepared. Each block was 10 m × 10 m. Within each block at each site, three main plots measuring 2 m × 2 m were randomly developed and equidistant. All plots were placed 2 m inside the block and plots were situated with distance of 2 m from each other. Three plots represented three treatments (i.e. control (no action), selective cutting and selective uprooting). The layout of the main plots at the six blocks in two sites is shown in Suppl. material 1: fig. S2). Selective cutting and uprooting involved the removal of individuals of *A. trifida* only, with minimal disturbance to other vegetation. In the case of cutting, all *A. trifida* individuals were removed by manual cutting using secateurs (pruning shears) and all other plant individuals were left intact. Individuals of *A. trifida* were cut from the lowest part of the plant stem (right above the ground). For the uprooting treatment, the entire plant, including roots, was uprooted or pulled out from the plots. As *A. trifida* has a fibrous root system, manual uprooting was easy. Any disturbance to original native plant communities, such as touching them, was minimised. All removal treatments were applied on 20 July 2022 (i.e. shortly before blooming). Within each main plot of 2 m × 2 m, four subplots measuring 1 m × 1 m were prepared for four sub-treatments. Three subplots were sown with three different seed mixtures and the remaining unsown subplot was used as the control. In seed mixture 1 (SM1), the seeds of three annual species (*Lactuca indica*, *Elsholtzia splendens* and *Portulaca oleracea*) were sown at 200 viable seeds of each species per subplot (i.e. 600 seeds m^{-2} per subplot). A similar method was followed for sowing seed mixture 2 (SM2) including three non-woody perennials (*Phragmites australis*, *Pennisetum alopecuroides* and *Plantago asiatica*) and seed mixture 3 (SM3) with three woody perennials (*Lespedeza bicolor*, *L. juncea* and *Sorbaria sorbifolia*). Seeds in each mixture were broadcast very close to the ground. Each seeded subplot was watered with a 15 L plastic watering pot. Manual watering of each subplot was done only at this stage. The application of seed mixtures was completed between 3 and 4 March 2023.

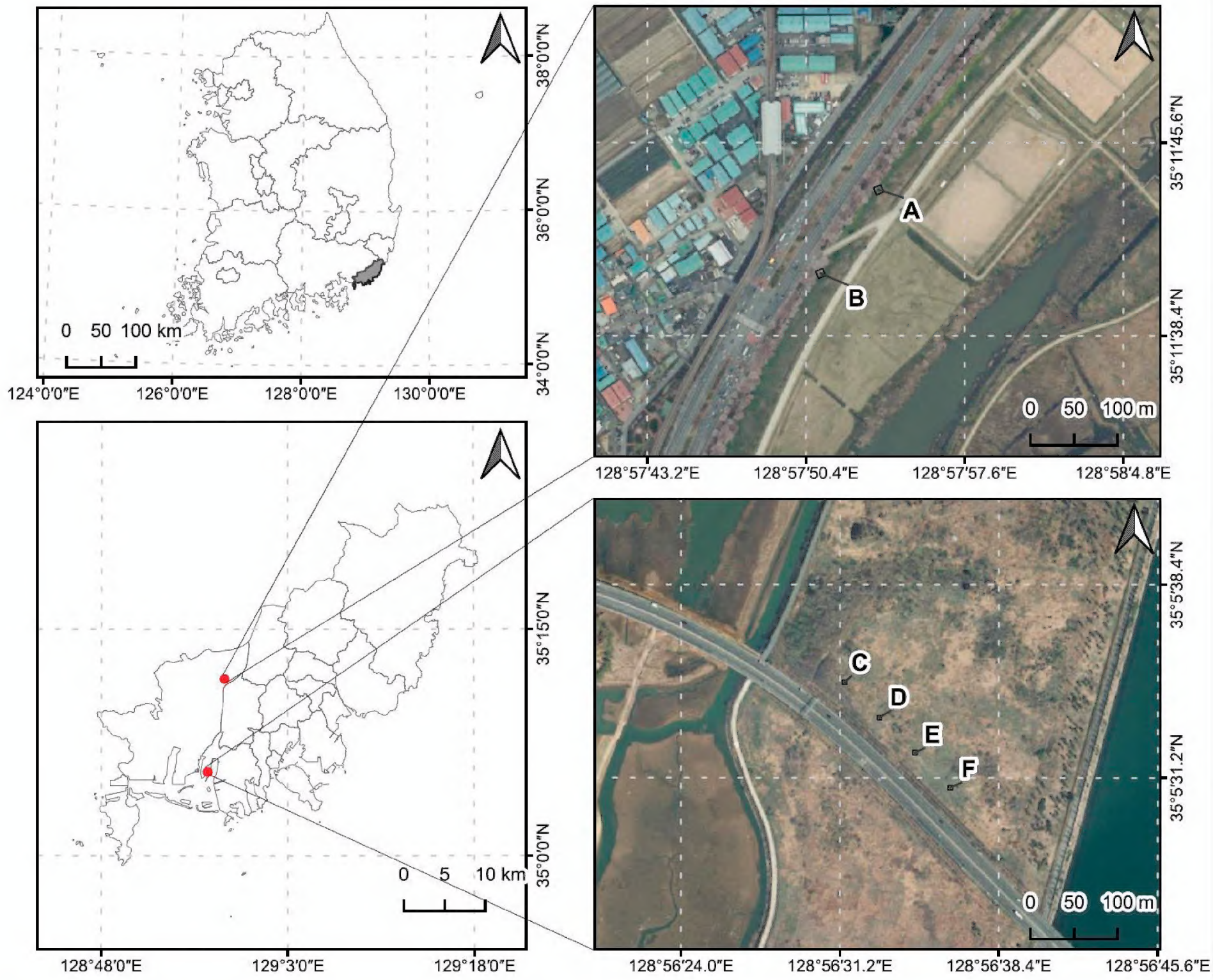

Figure 1. Map of the study sites and aerial view of blocks. Blocks A, B are located at site 1 and blocks C–F are located at site 2 in Busan, Republic of Korea.

Species selection and functional classification

Nine native species from three functional groups, annual (*Lactuca indica*, *Elsholtzia splendens* and *Portulaca oleracea*), non-woody perennials (*Phragmites australis*, *Pennisetum alopecuroides* and *Plantago asiatica*) and woody perennials (*Lespedeza bicolor*, *Lespedeza juncea* and *Sorbaria sorbifolia*) were identified for the current experiment. These native species were selected, based on their ability to suppress invasive plants in previous pot and field scale studies (Byun and Lee 2018; Byun et al. 2020b; Byun 2023), availability of their seeds in the market, their high germination rates, their distribution (native to S. Korea) and their ability to maximise functional diversity. The plant nomenclature used in this paper is the same as that used in the PLANT List (http://www.theplantlist.org), which lists accepted names and Flora of Korea Editorial Committee (2007).

Seed preparation

Seeds of native plants were purchased from authorised seed suppliers (in winter 2022). Seeds were obtained from multiple suppliers, because they could not be obtained from a single supplier. Seed suppliers included Aram Seeds (Seoul, Republic

of Korea), Xplant (Seoul, Republic of Korea) and New Korea Farm (Seongnam, Republic of Korea) and others. Seed viability was standardised by applying the same number of viable seeds per species (600 seeds m^{-2}) to experimental units. To determine the percentage of viable seeds, a germination test was conducted in the laboratory. All seeds were cold-stratified (6 months) at 3 °C before the germination test, following standard methods (Lindig-Cisneros and Zedler 2001). Then, 100 seeds per species were placed on three Petri dishes with Whatman® No. 1 filter paper moistened with 6 ml of distilled water under fluorescent light. Only species with high germination rates (above 5%) were used for the field experiment.

Measurements and statistical analyses

In August 2023, the number of shoots, plant height and plant cover of all species (including *A. trifida*) in each treatment and control plot were measured. For the number of shoots, we counted all shoots of each species in each plot manually. To determine plant cover, the percentage of each species was estimated using reference frames representing 50% and 25% of the total plot area. The main response variable was the importance value index, calculated based on the abundance of *A. trifida*. The importance value index (IVI) is a crucial metric in ecological studies, particularly when assessing the performance and impact of an invasive species (Byun et al. 2020b). IVI integrates multiple factors to provide a comprehensive picture of a species' dominance and ecological role within a community. IVI values were calculated by summing the relative plant cover (%) and the relative shoot density (%). This holistic approach is essential for understanding how invasive species perform relative to native species. High IVI values for an invasive species indicate that it is not only widespread, but also occupies a significant portion of the community's resources. This can highlight an invasive species' potential to outcompete and displace native species, altering community structure and ecosystem functions (Kohli et al. 2004). Additional main variables were the Shannon–Wiener diversity index (H') of all species (Keylock 2005) in each quadrat and invasive seed yield. The raw diversity index datasheets are included in Suppl. materail 2. To test the re-invasion potential of *A. trifida*, seeds were harvested on 9 November 2023 (all individuals were fully mature) in each subplot (Ferreras and Galetto 2010; Leskovsek et al. 2012; Goplen et al. 2016). All seeds of only *A. trifida* within each quadrat were manually detached by hands. The seed yield was measured as an indicator of the potential for invasion in the future. Seeds were oven-dried for 48 hours at 80 °C before measuring their yields in each treatment. Given that invasive seed yield in subplots did not show any specific pattern, data are not presented here.

ANOVA was used to evaluate the effects of various treatments on response variables. A generalised linear mixed model (REML; *F*-test) was used to account for the random block design (Bolker et al. 2009). In the ANOVA analysis model, main factors of treatments (different eradication methods for Fig. 2. and different seed mixtures within each eradication method for Fig. 3) and the site factors were included along with the random block effect. The three main response variables were: (1) the importance of the value index, calculated, based on the abundance of *A. trifida*, (2) the Shannon–Wiener diversity index (H') of all species in each quadrat and (3) invasive seed yield (g, log-transformed).

The normality of residuals and homoscedasticity were evaluated, and the response variables were transformed when necessary. Amongst the main response variables,

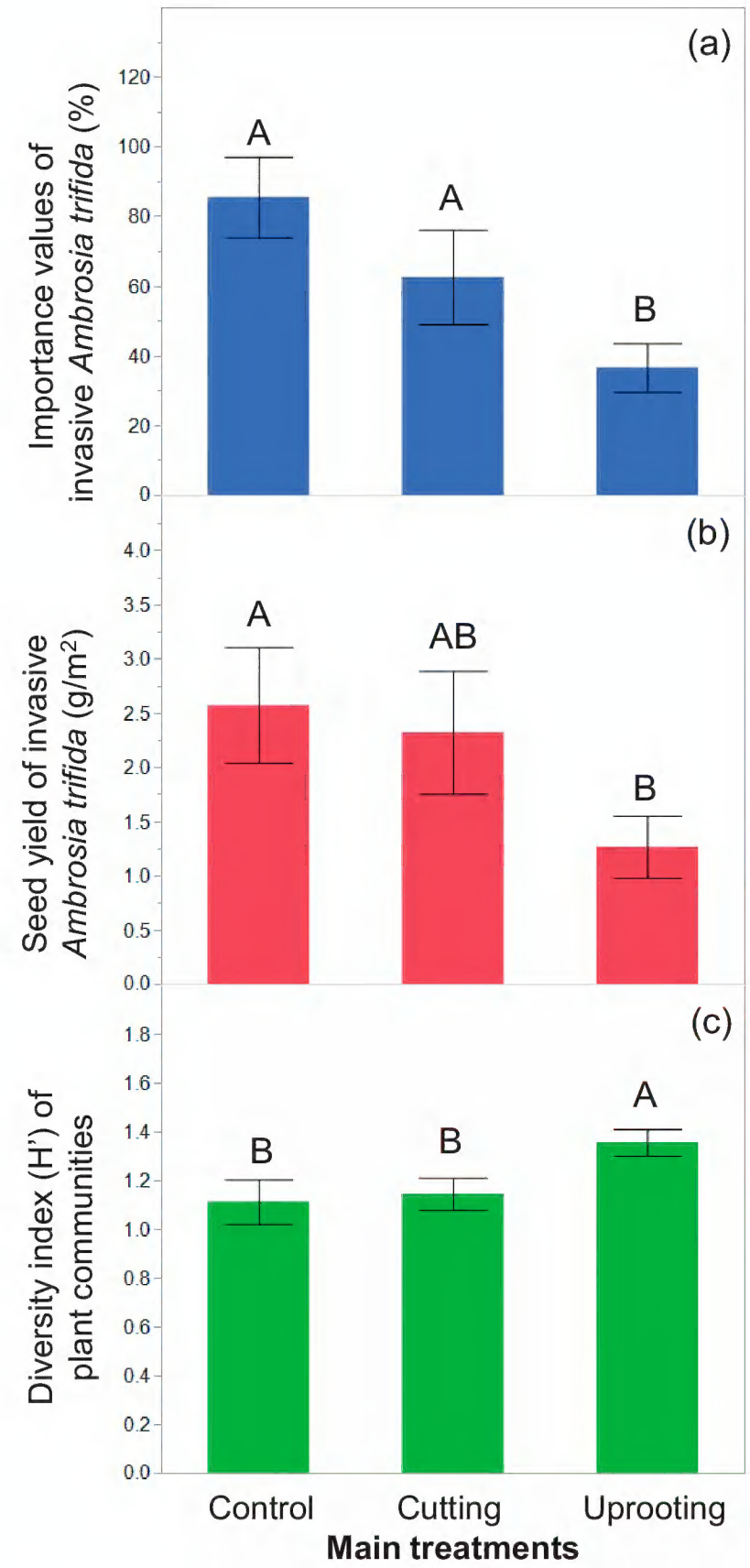

Figure 2. Effects of various eradication methods on *Ambrosia trifida* performance (importance values (**a**) and seed yield (**b**)) and plant community diversity (**c**). Error bars indicate standard errors. Bars with the same letters were not significantly different at $P < 0.05$.

only invasive seed yield (g) was log-transformed during analysis. When significant ($P < 0.05$) treatment effects were detected, *t*-tests were used to compare means of treatments. ANOVA and correlation analyses were conducted using JMP (SAS Institute Inc., Cary, NC, USA). Pearson correlation coefficients were calculated for *A. trifida* importance values and the diversity index (H') using data for 2023 in JMP.

Results

Response of invasive species *A. trifida* and native diversity to control measures

The effects of different control measures on the performance of *A. trifida* (Fig. 2a, b) and diversity of native plant communities (Fig. 2c), as well as the effects of sowing native seeds on invasion (Fig. 3a) and native diversity (Fig. 3b) were determined.

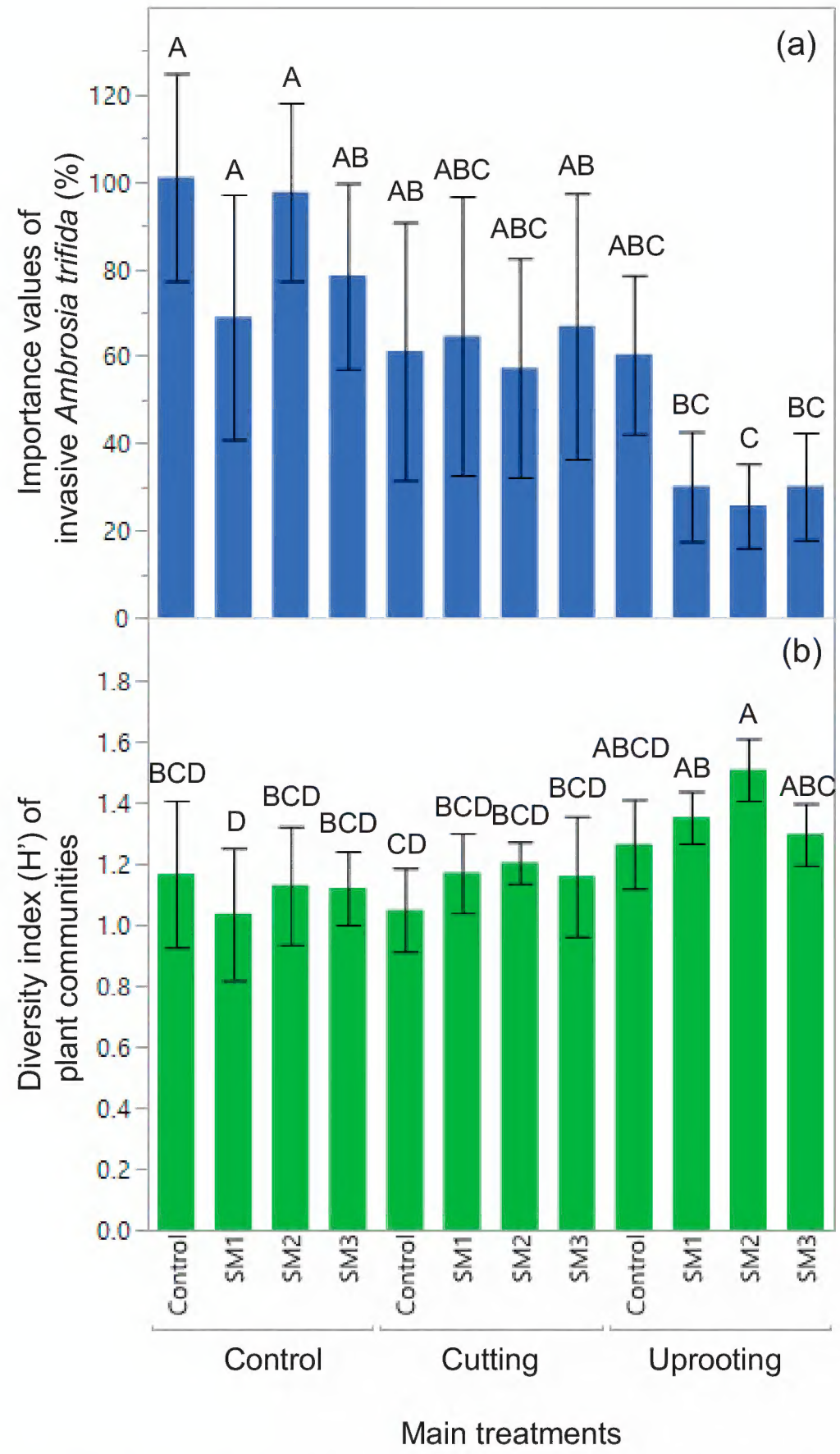

Figure 3. Effects of sub-treatments (sowing native seeds) within main treatments (eradication methods) on the importance values of *Ambrosia trifida* (**a**) and diversity of native plant communities (**b**). Error bars represent standard errors. Bars with the same letters were not significantly different. SM1 stands for seed mixture 1 (annuals), SM2 for non-woody perennials, SM3 for woody perennials.

The main eradication method had significant effects on abundance of *A. trifida* ($F_{2,50}$ = 9.5624; P = 0.0003) and the effect of the site factor was also significant ($F_{1,3}$ = 21.8288; P = 0.0167). The importance values for *A. trifida* were significantly lower in uprooting treatments than in control and cutting treatments (Fig. 2a). In addition, site 2 had much more abundant *A. trifida* populations than site 1 (as site factor). The seed yield of *A. trifida* was marginally affected by the control method ($F_{2,31}$ = 3.2787; P = 0.0511) (Fig. 2b). Similarly, the main eradication method had significant effects on native diversity ($F_{2,53}$ = 8.9271; P = 0.0005), but the effect of the site factor was not significant ($F_{1,3}$ = 7.2109; P = 0.0747). Uprooting significantly increased native diversity; however, cutting did not improve diversity over that in the control (Fig. 2c).

Response of *A. trifida* and native diversity to sowing native seeds following invasion control

Sowing native seeds did not have significant effects on the *A. trifida* importance value ($F_{9,41}$ = 0.7458; P = 0.6653), while the main eradication treatments had significant effects ($F_{2,41}$ = 9.2215; P = 0.0005) and the effect of the site factor was also significant ($F_{1,3}$ = 21.5360; P = 0.0168) in a nested experimental design. The importance values of *A. trifida* were much lower in the subplots with uprooting than in the control (no seed added) (Fig. 3a). Sowing native seeds did not influence the diversity index ($F_{9,44}$ = 0.7998; P = 0.6184), despite significant effects of eradication ($F_{2,44}$ = 8.6236; P = 0.0007) and the effect of the site factor was not significant ($F_{1,3}$ = 7.2109; P = 0.0747) in a nested experimental design. The diversity index was much higher in the subplots with uprooting than in the control (Fig. 3b).

Variations in plant cover of native species after eradication, but before sowing native species and after sowing seeds, were also observed (Table 1). Only four (one of each woody and annual and two of perennials) out of nine sown species were established. One of the most abundant native species was a woody species (*L. bicolor* Turcz.) (Table 1). However, this species was already observed in the community before sowing native seeds. The annual herb *L. indica* L. amongst the sown species was absent in the existing native vegetation and showed a many-fold increase in its cover (Table 1). There was little success in the establishment of the perennial grass *P. australis*. Three native annual herbs, *Acalypha australis*, *Commelina communis* and *Persicaria lapathifolia*, one annual legume, *Glycine soja* and one perennial herb, *Stellaria aquatica*, had appeared after eradiation of *A. trifida* and sowing native species. The establishment of these species after restoration initiatives, regardless of whether or not seeds of these plants were included in the seed mixture, is likely due to a decrease in competition for resources in the absence of *A. trifida*, a strong competitor, facilitative effects of companion native species and differences in the growth season of these species. *Humulus japonicus*, a perennial invasive climber, appeared immediately after the eradication of *A. trifida*, but its abundance decreased slightly after sowing native seeds. Another invasive annual herb, *Lactuca scariola*, was present before restoration initiatives began, but disappeared afterwards. One non-native annual herb, *Bidens pilosa*, a non-target species, also disappeared after eradication of *A. trifida* and sowing native species (Table 1). A significant negative correlation was observed between importance values of *A. trifida* and diversity index (Fig. 4).

Discussion

When invasive plants are partially removed, the effects on re-invasion can vary depending on the mode of reproduction of the invasive species, such as sexually (through seeds) and asexually (through rhizomes), as well as the persistence of the seed bank. For instance, re-sprouting from roots, rhizomes and plant stubs occurs in various invasive species (e.g. *Cyperus rotundus*, *Lantana camara*, *Phragmites australis* and *Rosa rugosa*), facilitating re-invasion after cutting or mowing aboveground plant parts. However, as observed in the current study on *A. trifida* (Fig. 2a), selective uprooting can significantly reduce the dominance of such invasive species. Similar results were found in a study of the sand dune invasive species *Rosa rugosa* (an invasive shrub in north-western

Table 1. Plant species and their cover before sowing native seeds (August 2022) and after sowing native seeds (September 2023). Species names in bold font were the sown species. PC, plant cover (%).

Species names	Growth habit	Native or not	Invasiveness	Sown species	PC 2022 (%)	PC 2023 (%)
***Lespedeza bicolor* Turcz.**[#]	**Perennial legume**	**Native**	**–**	**Sown**	**23.00**	**37.25**
Humulus japonicus Siebold & Zucc.	Perennial climber	Native	Invasive	–	46.12	36.80
Pueraria lobata Maesen S. M. Almeida ex Sanjappa & Predeep	Perennial vines	Native	–	–	28.37	34.00
Melothria japonica L.	Annual climber	Native	–	–	10.67	33.22
***Lactuca indica* L.**[#]	**Annual herb**	**Native**	**–**	**Sown**	**0**	**21.83**
Rubus parvifolius L.	Perennial shrub	Native	–	–	20.00	15.63
Acalypha australis L.	Annual herb	Native	–	–	0	14.50
Achyranthes bidentata var. *japonica* (Miq.) Nakai	Annual herb	Native	–	–	16.70	14.18
***Pennisetum alopecuroides* (L.) Spreng.**[#]	**Perennial grass**	**Native**	**–**	**Sown**	**17.00**	**13.33**
Commelina communis L.	Annual herb	Native	–	–	0	11.49
Artemisia indica Willd.	Annual herb	Native	–	–	2.50	10.75
Persicaria perfoliate (L.) H.Gross	Annual climbing	Native	–	–	7.00	7.57
Paederia foetida L.	Perennial herb	Native	–	–	18.5	7.20
Setaria viridis (L.) P. Beauv.	Perennial grass	Native	–	–	7.00	6.04
Cocculus trilobus (Thunb.) DC.	Climbing shrub	Native	–	–	8.25	5.50
Equisetum arvense L.	Perennial herb	Native	–	–	3.00	5.50
Stachys japonica L.	Perennial herb	Native	–	–	7.00	5.17
***Phragmites australis* (Cav.) Trin. ex Steud.**[#]	**Perennial grass**	**Native**	**–**	**Sown**	**0**	**4.75**
Glycine soja Siebold & Zucc.	Annual legume	Native	–	–	0	4.14
Artemisia lancea Van.	Perennial	Native	–	–	8.75	3.73
Digitaria ciliaris (Retz.) Koeler	Annual grass	Native	–	–	0.67	3.65
Persicaria lapathifolia (L.) Delarbre	Annual herb	Native	–	–	0	3
Stellaria aquatica (L.) Scop	Perennial herb	Native	–	–	0	1.25
Lactuca scariola L.	Annual herb	Non-native	Invasive	–	21.00	0
Fallopia dumetorum (L.) Holub	Annual climber	Native	–	–	1.00	0
Amphicarpaea bracteata edgeworthii Benth.	Annual climber	Native	–	–	6.67	0
Bidens pilosa L.	Annual herb	Non-native	–	–	20.00	0

[#] Seed mixtures of native species. Five sown native species (two annuals: *E. splendens* and *P. oleracea*; one non-woody perennial: *P. asiatica*; and two woody perennials: *L. juncea* and *S. sorbifolia*) did not become established.

Europe with negative effects on coastal biodiversity) (Kollmann et al. 2011), in which no re-sprouting was detected 2 months after uprooting. However, in the current study, uprooting reduced the abundance of *A. trifida* after 1 year. Mechanical control, including uprooting, cutting and ploughing, is an effective measure to eradicate *Ambrosia* species (Gerber et al. 2011). Generally, habitat disturbance caused by the mechanical removal of invasive plants facilitates re-invasion and hinders the recovery of native diversity. For instance, ploughing can significantly disrupt the soil environment, while mowing down all species can eliminate valuable native plants that resist *A. trifida* invasion. Selective uprooting, on the other hand, minimally impacts the habitat, preserving native vegetation and facilitating the eradication of invasive species *A. trifida*, ultimately benefitting native diversity. This expectation is supported by our results (Fig. 2c) demonstrating a significantly higher diversity of plant communities in uprooting plots than in control or cutting plots. The insignificant suppressive effect of

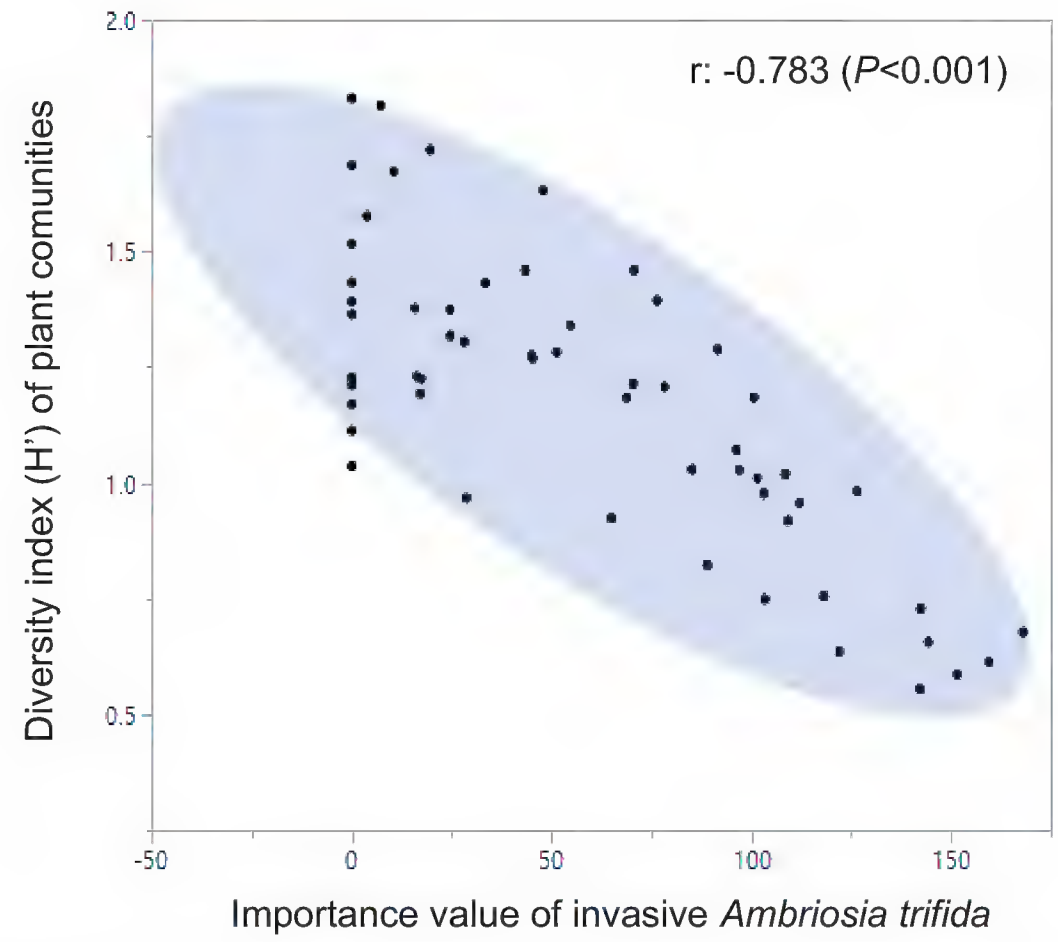

Figure 4. Relationship between the importance values of *Ambrosia trifida* and diversity index (H') of plant communities.

selective cutting in the current study is contrary to results of an earlier study in which a 77% reduction in *A. trifida* importance values was reported after cutting all plant species (Byun et al. 2020b). The lack of a significant reduction in *A. trifida* abundance in the current study might be explained by a difference in the extent of the disturbance (Fox 1979; Byun et al. 2020b). Although the selective nature of cutting in this study was not destructive to the habitat and other vegetation, re-sprouting from *A. trifida* plant stubs and belowground plant parts did not support an increase in native plant diversity in this treatment (Fig. 2c). This indicates that *A. trifida* has strong potential to regrow and flower from remaining plant parts at the removal site. In a previous study, after clipping aboveground stems of *A. artemisiifolia* four times in a growing season, > 67% of individuals survived and, amongst these, > 97% flowered (Patracchini et al. 2011). Our findings on the seed yield of *A. trifida* in cutting and uprooting treatments (Fig. 2b) were consistent with these earlier results.

The addition of native seeds following invasion control is an effective strategy for controlling re-invasion and increasing biodiversity (Bucharova and Krahulec 2020); however, this approach has not been investigated extensively (Kettenring and Adams 2011; Singh and Byun 2023) and the results of various studies have been contradictory (Patracchini et al. 2011; Byun and Lee 2018; Byun et al. 2020b; Dong et al. 2020a; Wang et al. 2022). The responses of invasive species and native diversity to sowing native seeds following invasion control can be neutral, positive (facilitation) or negative (suppression) depending on various factors, including the identity of invasive species, removal strategy (Flory and Clay 2009; Kollmann et al. 2011; Cutting and Hough-Goldstein 2013), diversity, density and features of seeds (Falk et al. 2013) and ecological conditions (Reinhardt Adams and Galatowitsch 2008). The responses of *A. trifida* invasive species to annual seed mix (SM1) were insignificant, but negative in control plots and significant, but positive in the cutting plots (Fig. 3). The suppressive effect of annuals on *A. trifida* abundance might be explained by the concept of limiting trait similarity (Yannelli et al. 2017), suggesting that interspecific

similarity (annuals in this case) leads to the competitive exclusion of invasive species (Macarthur and Levins 1967). Given that cutting did not reduce the abundance of *A. trifida* (Fig. 2) or increase diversity index (Fig. 3a), annuals might have facilitated further invasion in the cutting plots. A consistent increase in invasive seeds and decrease in native seeds at invaded sites can also contribute to *A. trifida* invasion and inhibit diversity recovery. A decrease of up to 83% in native seeds was reported within 8 years of *A. trifida* invasion (Wang et al. 2022). We detected the suppression of *A. trifida* invasive species in response to sowing native seeds in selective uprooting plots; however, the magnitude of the effect was similar for seed mixtures including annual and woody species (Fig. 3a). The perennial seed mix following uprooting had a greater suppressive effect on *A. trifida* invasion than those of both annual and woody seed mixes. Thus, the eradication of *A. trifida* by uprooting and follow-up restoration by sowing native seeds can be an effective strategy to protect against its invasion. The lower seed yield of *A. trifida* in uprooting plots supports this assumption.

The variations in plant cover of native vegetation between 2022 and 2023 can be attributed to the combined effects of invasive species removal, sowing of native species, differences in growth habits, improved environmental conditions and interspecific interactions. The eradication of *A. trifida* may have reduced competition for resources, such as light, water and nutrients and, thereby, allowed other species to flourish. The sowing mixtures of native species would be expected to directly increase the presence and cover of these species. This is evident from the appearance of species that were absent in 2022, such as *L. indica* and *P. australis* in 2023. Annual species such as *L. indica* and *A. australis* can quickly colonise and cover ground within a single growing season. Perennials, on the other hand, might show more substantial growth over several years. This explains why some annual species were completely absent in 2022 and appeared in 2023 after sowing, while some perennials maintained or slightly increased their cover. For example, *M. japonica* increased its cover from 10.67% to 33.22%, indicating a competitive advantage or favourable conditions for this species post-eradication. Likewise, the increase in cover of *L. bicolor* from 23.00% to 37.25%, might be due to reduced competition and to its being a sown species.

While native seed sowing suppressed *A. trifida* invasion to different extents in each treatment, it did not impact the recovery of native diversity significantly. Sowing native seeds following invasion removal has been reported not to be a promising strategy for increasing native plant biodiversity, as reported in recent studies of *A. trifida* (Bucharova and Krahulec 2020; Byun et al. 2020b; Nagy et al. 2022) and other invasive plant species (Dickson and Busby 2009; Cutting and Hough-Goldstein 2013; Tarsa et al. 2022). In addition, there were no differences amongst seeds of different functional groups with respect to native diversity recovery, regardless of the method employed for *A. trifida* invasion eradication (Fig. 3c). A recent greenhouse study also reported that the diverse native species of different functional groups does not suppress *A. trifida* (Byun and Lee 2018). However, the early arrival of native species strongly contained *A. trifida* invasion. There are multiple explanations for the lack of recovery of native diversity after seed addition: (1) Field conditions may not be favourable for seed germination; (2) An important factor in restoring native diversity using native seeds is appropriate seed density. Considering the harsh conditions and low germination rate, 600 seeds m^{-2} may not be a sufficient density (Burton et al. 2006; Reinhardt Adams and Galatowitsch 2008; Byun and Lee 2018; Shaw et al. 2020; Byun et al. 2020b; Tarsa et al. 2022); however, insignificant recovery with a high seed density has also been reported (Dickson

and Busby 2009); (3) A narrow niche range of some of the native species did not support their dominance in current distinct habitat conditions (Pulliam 2000); (4) Seeds of perennials, such as *P. australis* and *P. alopecuroides*, were hairy (Suppl. material 1: fig. S1) and hardly encountered the ground or soil directly during sowing; it is, therefore, possible that these species could not grow and contribute to native diversity; (5) Commercial sources of seeds, generally agronomically propagated (Höfner et al. 2022), may contribute to low adaptability to targeted local conditions (Bucharova et al. 2019); (6) Sowing seeds of one functional group in each plot might not exert strong effects on diversity. More diversified functional groups or types of native seeds in seed mixtures are needed to better assess whether seeding native species counteracts invasive plants and restores native biodiversity; (7) *A. trifida* may be too difficult to control via native species (Byun and Lee 2018). Given that uprooting reduced invasion substantially, consistently uprooting *A. trifida* for a longer period, at least 3 years (soil seed-bank densities decreased by > 99% in 2 years (Dong et al. 2020b)), in a larger area to limit the arrival of invasive seeds and adding native seeds can facilitate the recovery of native diversity. We did not test the role of functional diversity; therefore, further studies should evaluate whether sowing seeds of all functional groups in combination restores native diversity.

Limitations of study

A major limitation of this study was relatively short monitoring time (1 year or less) after restoration. We think that longer monitoring would have yielded better results. The short monitoring time may explain, at least partly, why sowing native seeds did not bring any additional benefit to the control of, or resistance to, *A. trifida* invasion. The seed mixtures of three functional groups of native plant species were employed: annuals, non-woody perennials and woody perennials, because we wanted to determine which functional group was most effective in providing biotic resistance to invasion in the year following eradication of *A. trifida*. Annuals were expected to perform better as they are usually fast-growing and become established in the first year after eradication. This is also expected, based on the limiting similarity hypothesis (*A. trifida* is also an annual plant species). However, we did not find any difference in biotic resistance to invasion between the functional groups of seeds; in fact, there was no difference between sowing and not sowing seeds. We only monitored plots soon after eradication because we considered one year as the critical window for invasive species re-invasion. If invasive species are not controlled within this short time frame, then it will be difficult to stop re-establishment of the invasive species afterwards. As we were acutely aware of the limited timeframe of this study, we ended up measuring the invasive seed yield as an indicator for potential future re-invasion after one year of monitoring.

One of the critical aspects of restoring native species using native seeds is the seed density. For instance, 600 pure live seeds (after considering germination rates per species) m^{-2} per subplot were sown. Originally, this density was considered sufficient in the initial experimental design, but under actual heterogeneous field conditions, many different factors can influence seeding efficiency. For example, the characteristics of experimental sites might not match the ecological niches of the restorative native species. In addition, it is also likely that seed density is reduced by their ingestion by some animals, such as birds, in the Spring. Considering these field limitations, we now consider that 10-fold higher seed density

would have been required to obtain meaningful and significant results; in fact, this density was recommended by a seed-based restoration workshop at a conference of the Society of Ecological Restoration (SER).

Implications for practice

The findings of this study have strong implications for the management of invasive plants and recovery of native plant diversity: (1) Cutting to eradicate plant invasion can result in wasted effort and resources, particularly if the targeted species can regrow or re-sprout from remaining plant parts. In the current study, cutting was selective and resulted in minimal disturbances of native vegetation. However, invasion was suppressed to only a small extent with insignificant differences between cutting plots and the control plot. Complete and destructive cutting of the entire vegetation may further increase invasion by reducing native plant diversity; (2) Selective uprooting is a promising tool for invasive plant management. Complete removal of invasive species from invaded communities and ecosystems will reduce competition pressure on native species for space, light and nutrients and form invasive propagules (roots, rhizomes, seeds etc.) and increase the performance (germination, establishment and diversity) of native communities; (3) Sowing seeds of diverse species following removal of the invasive species is critical for the rapid recovery of native diversity; (4) Compared with selective uprooting, mowing of all species is not an effective strategy for the management of invasive plants because it does not leave any native species to resist re-invasion; (5) Although this study was conducted solely at two field sites within the Republic of Korea, our findings can readily be extrapolated to other countries. This generalisability stems from the underlying ecological principles uncovered, namely, the importance of leaving no propagule behind for achieving effective eradication outcomes. This fundamental principle is relevant, irrespective of geographic context, making it applicable across diverse regions; (6) Lastly, it is important to consider the potential environmental or ecological side effects of selective uprooting. For instance, hand-pulling to uproot all invasive plants can slightly disturb soil composition, potentially impacting soil microorganism communities and the legacy effects of soil on biogeochemical processes. Therefore, selective uprooting must be executed with meticulous care to minimise disturbances to the soil surface and other native species.

Conclusions

This study concludes that selective uprooting is a more effective tool than cutting for suppressing *A. trifida* invasion and increasing the diversity of native plant communities. The findings of this study support the expectation that uprooting of invasive species before flowering with minimum habitat disturbances can immediately reduce competition for remaining native species and concurrently can increase native diversity in the next growing season due to decreases in the number of seeds of *A. trifida*, the invasive species. Suppression of plant invasion further increased after sowing native seeds; however, this was only valid if the invasive plant was eradicated by uprooting. Therefore, sowing seeds to restore native diversity at sites where *A. trifida* invasion has been eradicated by cutting may result in the waste of native seeds, time and other resources. Sowing native seeds after removal of *A. trifida* by cutting and uprooting facilitated recovery of native diversity; however, uprooting followed by sowing native plants was more effective.

Acknowledgements

We thank Dr. Ho Choi and Dr. Norul Sobuj for their assistance with the fieldwork. We also thank Dr. Minwoo Oh for drawing the study site map in QGIS. We thank Eulsukdo Ecological Park Nakdonggang Estuary Eco Center for field cooperation.

Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

Funding

This work was supported by the National Research Foundation of Korea (NRF) grant funded by the Korea government (MSIT) (2022R1A2C1003504). This study was supported by a joint research project from the Ministry of Environment, Republic of Korea (Project number: 2021002270004).

Author contributions

CB conceptualised and designed the study; CB, KS and JL collected data, CB analysed and constructed graphs and tables; CB, KS and JL wrote the manuscript; SHH, TKY and HK reviewed and edited the manuscript; CB and SHH engaged in project collaboration.

Author ORCIDs

Chaeho Byun https://orcid.org/0000-0003-3209-3275
Kripal Singh https://orcid.org/0000-0003-2845-7160
Sun Hee Hong https://orcid.org/0000-0001-7581-0604
Jangho Lee https://orcid.org/0009-0006-3274-6378
Tae Kyung Yoon https://orcid.org/0000-0003-0627-0135
Hojeong Kang https://orcid.org/0000-0002-2088-6406

Data availability

Dataset was shared in the open access file directories of Figshare. https://doi.org/10.6084/m9.figshare.26425879.

References

Abul-Fatih H, Bazzaz F (1979) The biology of *Ambrosia trifida* L. II. Germination, emergence, growth and survival. The New Phytologist 83(3): 817–827. https://doi.org/10.1111/j.1469-8137.1979.tb02313.x

Adams SN, Jennings S, Warnock N (2020) Plant invasion depresses native species richness, but control of invasive species does little to restore it. Plant Ecology & Diversity 13(3–4): 257–266. https://doi.org/10.1080/17550874.2020.1817998

Adhikari P, Lee YH, Poudel A, Hong SH, Park YS (2023) Global spatial distribution of *Chromolaena odorata* habitat under climate change: Random forest modeling of one of the 100 worst invasive alien species. Scientific Reports 13(1): 9745. https://doi.org/10.1038/s41598-023-36358-z

Bolker BM, Brooks ME, Clark CJ, Geange SW, Poulsen JR, Stevens MH, White JS (2009) Generalized linear mixed models: A practical guide for ecology and evolution. Trends in Ecology & Evolution 24(3): 127–135. https://doi.org/10.1016/j.tree.2008.10.008

Boxriker M, Privett SDJ, Richardson DM, Gaertner M (2022) Active restoration in South African fynbos–A long-term perspective from the Agulhas Plain. Transactions of the Royal Society of South Africa 77(2): 133–143. https://doi.org/10.1080/0035919X.2022.2087116

Bucharova A, Krahulec F (2020) Native seed addition as an effective tool for post-invasion restoration. Basic and Applied Ecology 42: 54–61. https://doi.org/10.1016/j.baae.2020.01.002

Bucharova A, Bossdorf O, Hölzel N, Kollmann J, Prasse R, Durka W (2019) Mix and match: Regional admixture provenancing strikes a balance among different seed-sourcing strategies for ecological restoration. Conservation Genetics 20(1): 7–17. https://doi.org/10.1007/s10592-018-1067-6

Burton CM, Burton PJ, Hebda R, Turner NJ (2006) Determining the optimal sowing density for a mixture of native plants used to revegetate degraded ecosystems. Restoration Ecology 14(3): 379–390. https://doi.org/10.1111/j.1526-100X.2006.00146.x

Byun C (2023) Role of priority effects in invasive plant species management: Early arrival of native seeds guarantees the containment of invasion by Giant ragweed. Ecology and Evolution 13(3): e9940. https://doi.org/10.1002/ece3.9940

Byun C, Lee EJ (2018) Giant ragweed invasion is not well controlled by biotic resistance. Journal of Plant Biology 61(5): 301–308. https://doi.org/10.1007/s12374-017-0472-6

Byun C, de Blois S, Brisson J (2013) Plant functional group identity and diversity determine biotic resistance to invasion by an exotic grass. Journal of Ecology 101(1): 128–139. https://doi.org/10.1111/1365-2745.12016

Byun C, Choi H, Kang H (2020a) Effects of cutting and sowing seeds of native species on giant ragweed invasion and plant diversity in a field experiment. Journal of Ecology and Environment 44(1): 1–8. https://doi.org/10.1186/s41610-020-00173-8

Byun C, Oh M, Lee EJ, Kang H (2020b) Seed density is as important as limiting similarity, diversity effect, and propagule pressure in plant restoration to control invasion. Ecological Engineering 144: 105712. https://doi.org/10.1016/j.ecoleng.2019.105712

Carlson AM, Gorchov DL (2004) Effects of herbicide on the invasive biennial *Alliaria petiolata* (garlic mustard) and initial responses of native plants in a southwestern Ohio forest. Restoration Ecology 12(4): 559–567. https://doi.org/10.1111/j.1061-2971.2004.00373.x

Castro-Díez P, Vaz AS, Silva JS, Van Loo M, Alonso Á, Aponte C, Bayón Á, Bellingham PJ, Chiuffo MC, DiManno N, Julian K, Kandert S, La Porta N, Marchante H, Maule HG, Mayfield MM, Metcalfe D, Monteverdi MC, Núñez MA, Ostertag R, Parker IM, Peltzer DA, Potgieter LJ, Raymundo M, Rayome D, Reisman-Berman O, Richardson DM, Roos RE, Saldaña A, Shackleton RT, Torres A, Trudgen M, Urban J, Vicente JR, Vilà M, Ylioja T, Zenni RD, Godoy O (2019) Global effects of non-native tree species on multiple ecosystem services. Biological Reviews of the Cambridge Philosophical Society 94(4): 1477–1501. https://doi.org/10.1111/brv.12511

Chauvel B, Fried G, Follak S, Chapman D, Kulakova Y, Le Bourgeois T, Marisavljevic D, Monty A, Rossi J-P, Starfinger U, Tanner R, Tassus X, Van Valkenburg J, Regnier E (2021) Monographs on invasive plants in Europe N° 5: *Ambrosia trifida* L. Botany Letters 168(2): 167–190. https://doi.org/10.1080/23818107.2021.1879674

Chen S, Bai X, Ye J, Chen W, Xu G (2024) Prediction of suitable habitat of alien invasive plant *Ambrosia trifida* in Northeast China under various climatic scenarios. Diversity 16(6): 322. https://doi.org/10.3390/d16060322

Chenot J, Affre L, Gros R, Dubois L, Malecki S, Passetti A, Aboucaya A, Buisson E (2018) Eradication of invasive *Carpobrotus* sp.: Effects on soil and vegetation. Restoration Ecology 26(1): 106–113. https://doi.org/10.1111/rec.12538

Clewley GD, Eschen R, Shaw RH, Wright DJ (2012) The effectiveness of classical biological control of invasive plants. Journal of Applied Ecology 49(6): 1287–1295. https://doi.org/10.1111/j.1365-2664.2012.02209.x

Courkamp JS, Meiman PJ, Nissen SJ (2022) Indaziflam reduces downy brome (*Bromus tectorum*) density and cover five years after treatment in sagebrush-grasslands with no impact on perennial grass cover. Invasive Plant Science and Management 15(3): 122–132. https://doi.org/10.1017/inp.2022.21

Cutting KJ, Hough-Goldstein J (2013) Integration of biological control and native seeding to restore invaded plant communities. Restoration Ecology 21(5): 648–655. https://doi.org/10.1111/j.1526-100X.2012.00936.x

David AS, Lake EC (2023) Biological control of vines: A review of past efforts, evaluation, and future directions. Biological Control 105257: 105257. https://doi.org/10.1016/j.biocontrol.2023.105257

Dickson TL, Busby WH (2009) Forb species establishment increases with decreased grass seeding density and with increased forb seeding density in a Northeast Kansas, USA, experimental prairie restoration. Restoration Ecology 17(5): 597–605. https://doi.org/10.1111/j.1526-100X.2008.00427.x

Donaldson R, Germino MJ (2022) Intra-site sources of restoration variability in severely invaded rangeland: Strong temporal effects of herbicide–weather interactions; weak spatial effects of plant community patch type and litter. Ecological Solutions and Evidence 3(3): e12172. https://doi.org/10.1002/2688-8319.12172

Dong H, Song Z, Liu T, Liu Z, Liu Y, Chen B, Ma Q, Li Z (2020a) Causes of differences in the distribution of the invasive plants *Ambrosia artemisiifolia* and *Ambrosia trifida* in the Yili Valley, China. Ecology and Evolution 10(23): 13122–13133. https://doi.org/10.1002/ece3.6902

Dong H, Liu T, Liu Z, Song Z (2020b) Fate of the soil seed bank of giant ragweed and its significance in preventing and controlling its invasion in grasslands. Ecology and Evolution 10(11): 4854–4866. https://doi.org/10.1002/ece3.6238

Enloe SF, Ditomaso JM, Orloff SB, Drake DJ (2005) Perennial grass establishment integrated with clopyralid treatment for yellow starthistle management on annual range. Weed Technology 19(1): 94–101. https://doi.org/10.1614/WT-03-262R1

Falk AD, Fulbright TE, Smith FS, Brennan LA, Ortega-Santos AJ, Benn S (2013) Does seeding a locally adapted native mixture inhibit ingress by exotic plants? Restoration Ecology 21(4): 474–480. https://doi.org/10.1111/j.1526-100X.2012.00918.x

Farmilo BJ, Moxham C (2023) The effectiveness of weed control in a threatened plant community: A grassland case study. Ecological Engineering 193: 107017. https://doi.org/10.1016/j.ecoleng.2023.107017

Ferreras AE, Galetto L (2010) From seed production to seedling establishment: Important steps in an invasive process. Acta Oecologica 36(2): 211–218. https://doi.org/10.1016/j.actao.2009.12.005

Flora of Korea Editorial Committee (2007) The general vascular plants of Korea. Academy Publishing Co, Seoul.

Flory SL, Clay K (2009) Invasive plant removal method determines native plant community responses. Journal of Applied Ecology 46(2): 434–442. https://doi.org/10.1111/j.1365-2664.2009.01610.x

Fox JF (1979) Intermediate-disturbance hypothesis. Science 204(4399): 1344–1345. https://doi.org/10.1126/science.204.4399.1344

Gerber E, Schaffner U, Gassmann A, Hinz H, Seier M, Müller-Schärer H (2011) Prospects for biological control of *Ambrosia artemisiifolia* in Europe: Learning from the past. Weed Research 51(6): 559–573. https://doi.org/10.1111/j.1365-3180.2011.00879.x

Goplen JJ, Sheaffer CC, Becker RL, Coulter JA, Breitenbach FR, Behnken LM, Johnson GA, Gunsolus JL (2016) Giant ragweed (*Ambrosia trifida*) seed production and retention in soybean and field margins. Weed Technology 30(1): 246–253. https://doi.org/10.1614/WT-D-15-00116.1

Höfner J, Klein-Raufhake T, Lampei C, Mudrak O, Bucharova A, Durka W (2022) Populations restored using regional seed are genetically diverse and similar to natural populations in the region. Journal of Applied Ecology 59(9): 2234–2244. https://doi.org/10.1111/1365-2664.14067

Jaureguiberry P, Titeux N, Wiemers M, Bowler DE, Coscieme L, Golden AS, Guerra CA, Jacob U, Takahashi Y, Settele J, Díaz S, Molnár Z, Purvis A (2022) The direct drivers of recent global anthropogenic biodiversity loss. Science Advances 8(45): eabm9982. https://doi.org/10.1126/sciadv.abm9982

Johnson DP, Catford JA, Driscoll DA, Gibbons P (2018) Seed addition and biomass removal key to restoring native forbs in degraded temperate grassland. Applied Vegetation Science 21(2): 219–228. https://doi.org/10.1111/avsc.12352

Kettenring KM, Adams CR (2011) Lessons learned from invasive plant control experiments: A systematic review and meta-analysis. Journal of Applied Ecology 48(4): 970–979. https://doi.org/10.1111/j.1365-2664.2011.01979.x

Keylock C (2005) Simpson diversity and the Shannon–Wiener index as special cases of a generalized entropy. Oikos 109(1): 203–207. https://doi.org/10.1111/j.0030-1299.2005.13735.x

Kim KD (2017) Distribution and management of the invasive exotic species *Ambrosia trifida* and *Sicyos angulatus* in the Seoul metropolitan area. Journal of Ecological Engineering 18(5): 27–36. https://doi.org/10.12911/22998993/76216

Kohli RK, Dogra KS, Batish DR, Singh HP (2004) Impact of invasive plants on the structure and composition of natural vegetation of northwestern Indian Himalayas. Weed Technology 18(sp1): 1296–1300. https://doi.org/10.1614/0890-037X(2004)018[1296:IOIPOT]2.0.CO;2

Kollmann J, Brink-Jensen K, Frandsen SI, Hansen MK (2011) Uprooting and burial of invasive alien plants: A new tool in coastal restoration? Restoration Ecology 19(3): 371–378. https://doi.org/10.1111/j.1526-100X.2009.00569.x

Kouame KB-J, Butts TR, Werle R, Johnson WG (2023) Impact of volatility reduction agents on dicamba and glyphosate spray solution pH, droplet dynamics, and weed control. Pest Management Science 79(2): 857–869. https://doi.org/10.1002/ps.7258

Laforest M, Martin SL, Bisaillon K, Soufiane B, Meloche S, Tardif FJ, Page E (2024) The ancestral karyotype of the Heliantheae Alliance, herbicide resistance, and human allergens: Insights from the genomes of common and giant ragweed. The Plant Genome 20442(2): e20442. https://doi.org/10.1002/tpg2.20442

Lee C-S, Cho Y-C, Shin H-C, Kim G-S, Pi J-H (2010) Control of an invasive alien species, *Ambrosia trifida* with restoration by introducing willows as a typical riparian vegetation. Journal of Ecology and Environment 33(2): 157–164. https://doi.org/10.5141/JEFB.2010.33.2.157

Leskovsek R, Datta A, Simoncic A, Knezevic SZ (2012) Influence of nitrogen and plant density on the growth and seed production of common ragweed (*Ambrosia artemisiifolia* L.). Journal of Pest Science 85(4): 527–539. https://doi.org/10.1007/s10340-012-0433-2

Levine JM, Adler PB, Yelenik SG (2004) A meta-analysis of biotic resistance to exotic plant invasions. Ecology Letters 7(10): 975–989. https://doi.org/10.1111/j.1461-0248.2004.00657.x

Li H, Kang Z, Hua J, Feng Y, Luo S (2022) Root exudate sesquiterpenoids from the invasive weed *Ambrosia trifida* regulate rhizospheric Proteobacteria. The Science of the Total Environment 834: 155263. https://doi.org/10.1016/j.scitotenv.2022.155263

Lindig-Cisneros R, Zedler J (2001) Effect of light on seed germination in *Phalaris arundinacea* L. (reed canary grass). Plant Ecology 155(1): 75–78. https://doi.org/10.1023/A:1013224514980

Ling X-J, Zhou Y-J, Yang Y-S, Xu Z-Q, Wang Y, Sun J-L, Zhu Y, Wei J-F (2022) A new cysteine protease allergen from *Ambrosia trifida* pollen: Proforms and mature forms. Molecular Immunology 147: 170–179. https://doi.org/10.1016/j.molimm.2022.05.003

Macarthur R, Levins R (1967) The limiting similarity, convergence, and divergence of coexisting species. American Naturalist 101(921): 377–385. https://doi.org/10.1086/282505

Montagnani C, Gentili R, Smith M, Guarino M, Citterio S (2017) The worldwide spread, success, and impact of ragweed (*Ambrosia* spp.). Critical Reviews in Plant Sciences 36(3): 139–178. https://doi.org/10.1080/07352689.2017.1360112

Nagy DU, Rauschert ES, Callaway RM, Henn T, Filep R, Pal RW (2022) Intense mowing management suppresses invader, but shifts competitive resistance by a native to facilitation. Restoration Ecology 30(1): e13483. https://doi.org/10.1111/rec.13483

O'Donnell J, Fryirs KA, Leishman MR (2016) Seed banks as a source of vegetation regeneration to support the recovery of degraded rivers: A comparison of river reaches of varying condition. The Science of the Total Environment 542: 591–602. https://doi.org/10.1016/j.scitotenv.2015.10.118

Park M, Park J, Hong S, Shin H (2012) First Asian report of leaf spot of *Ambrosia trifida* caused by *Septoria epambrosiae*. Plant Disease 96(2): 289–289. https://doi.org/10.1094/PDIS-10-11-0845

Park HJ, Hong MG, Kim JG (2020) Effects of soil fertility and flooding regime on the growth of *Ambrosia trifida*. Landscape and Ecological Engineering 16(1): 39–46. https://doi.org/10.1007/s11355-019-00403-9

Patracchini C, Vidotto F, Ferrero A (2011) Common ragweed (*Ambrosia artemisiifolia*) growth as affected by plant density and clipping. Weed Technology 25(2): 268–276. https://doi.org/10.1614/WT-D-09-00070.1

Pearson DE, Ortega YK, Runyon JB, Butler JL (2016) Secondary invasion: The bane of weed management. Biological Conservation 197: 8–17. https://doi.org/10.1016/j.biocon.2016.02.029

Pearson DE, Ortega YK, Eren Ö, Hierro JL (2018) Community assembly theory as a framework for biological invasions. Trends in Ecology & Evolution 33(5): 313–325. https://doi.org/10.1016/j.tree.2018.03.002

Peng W, Lam SS, Sonne C (2020) Support Austria's glyphosate ban. Science 367(6475): 257–258. https://doi.org/10.1126/science.aba5642

Pergl J, Härtel H, Pyšek P, Stejskal R (2020) Don't throw the baby out with the bathwater–ban of glyphosate use depends on context. NeoBiota 56: 27–29. https://doi.org/10.3897/neobiota.56.51823

Pickart AJ, Miller LM, Duebendorfer TE (1998a) Yellow bush lupine invasion in northern California coastal dunes I. Ecological impacts and manual restoration techniques. Restoration Ecology 6(1): 59–68. https://doi.org/10.1046/j.1526-100x.1998.00618.x

Pickart AJ, Theiss KC, Stauffer HB, Olsen GT (1998b) Yellow bush lupine invasion in Northern California Coastal Dunes II. Mechanical Restoration Techniques. Restoration Ecology 6(1): 69–74. https://doi.org/10.1046/j.1526-100x.1998.00619.x

Pulliam HR (2000) On the relationship between niche and distribution. Ecology Letters 3(4): 349–361. https://doi.org/10.1046/j.1461-0248.2000.00143.x

Pyšek P, Richardson DM (2010) Invasive species, environmental change and management, and health. Annual Review of Environment and Resources 35(1): 25–55. https://doi.org/10.1146/annurev-environ-033009-095548

Regnier EE, Harrison SK, Loux MM, Holloman C, Venkatesh R, Diekmann F, Taylor R, Ford RA, Stoltenberg DE, Hartzler RG, Davis AS, Schutte BJ, Cardina J, Mahoney KJ, Johnson WG (2016) Certified crop advisors' perceptions of giant ragweed (*Ambrosia trifida*) distribution, herbicide resistance, and management in the Corn Belt. Weed Science 64(2): 361–377. https://doi.org/10.1614/WS-D-15-00116.1

Reinhardt Adams C, Galatowitsch SM (2008) The transition from invasive species control to native species promotion and its dependence on seed density thresholds. Applied Vegetation Science 11(1): 131–138. https://doi.org/10.1111/j.1654-109X.2008.tb00211.x

Robichaud CD, Rooney RC (2021) Effective suppression of established invasive *Phragmites australis* leads to secondary invasion in a coastal marsh. Invasive Plant Science and Management 14(1): 9–19. https://doi.org/10.1017/inp.2021.2

Roy HE, Pauchard A, Stoett P, Renard Truong T, Bacher S, Galil BS, Hulme PE, Ikeda T, Sankaran KV, McGeoch MA (2023) IPBES Invasive Alien Species Assessment: Summary for Policymakers. https://doi.org/10.5281/zenodo.7430692

Schooler SS, Cook T, Prichard G, Yeates AG (2010) Disturbance-mediated competition: The interacting roles of inundation regime and mechanical and herbicidal control in determining native and invasive plant abundance. Biological Invasions 12(9): 3289–3298. https://doi.org/10.1007/s10530-010-9722-y

Shackleton CM, Ruwanza S, Sinasson Sanni GK, Bennett S, De Lacy P, Modipa R, Mtati N, Sachikonye M, Thondhlana G (2016) Unpacking Pandora's box: Understanding and categorising ecosystem disservices for environmental management and human wellbeing. Ecosystems (New York, N.Y.) 19(4): 587–600. https://doi.org/10.1007/s10021-015-9952-z

Shackleton RT, Biggs R, Richardson DM, Larson BM (2018) Social-ecological drivers and impacts of invasion-related regime shifts: Consequences for ecosystem services and human wellbeing. Environmental Science & Policy 89: 300–314. https://doi.org/10.1016/j.envsci.2018.08.005

Shaw N, Barak RS, Campbell RE, Kirmer A, Pedrini S, Dixon K, Frischie S (2020) Seed use in the field: Delivering seeds for restoration success. Restoration Ecology 28(S3): S276–S285. https://doi.org/10.1111/rec.13210

Sheley RL, James J (2010) Resistance of native plant functional groups to invasion by medusahead (*Taeniatherum caput-medusae*). Invasive Plant Science and Management 3(3): 294–300. https://doi.org/10.1614/IPSM-D-09-00056.1

Sheley RL, Mangold JM, Anderson JL (2006) Potential for successional theory to guide restoration of invasive-plant-dominated rangeland. Ecological Monographs 76(3): 365–379. https://doi.org/10.1890/0012-9615(2006)076[0365:PFSTTG]2.0.CO;2

Shen C, Chen P, Zhang K, He M, Wan J, Wang Y, Tao Z, Huang W, Siemann E (2023) Dynamics and mechanisms of secondary invasion following biological control of an invasive plant. The New Phytologist 238(6): 2594–2606. https://doi.org/10.1111/nph.18878

Silva TS, Arneson NJ, DeWerff RP, Smith DH, Silva DV, Werle R (2023) Preemergence herbicide premixes reduce the risk of soil residual weed control failure in corn. Weed Technology 37(4): 1–12. https://doi.org/10.1017/wet.2023.45

Simberloff D, Martin J-L, Genovesi P, Maris V, Wardle DA, Aronson J, Courchamp F, Galil B, García-Berthou E, Pascal M, Pyšek P, Sousa R, Tabacchi E, Vilà M (2013) Impacts of biological invasions: What's what and the way forward. Trends in Ecology & Evolution 28(1): 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Singh K, Byun C (2023) Ecological restoration after management of invasive alien plants. Ecological Engineering 197: 107122. https://doi.org/10.1016/j.ecoleng.2023.107122

Soltani N, Shropshire C, Sikkema P (2011) Giant ragweed (*Ambrosia trifida* L.) control in corn. Canadian Journal of Plant Science 91(3): 577–581. https://doi.org/10.4141/cjps2010-004

Tarsa EE, Holdaway BM, Kettenring KM (2022) Tipping the balance: The role of seed density, abiotic filters, and priority effects in seed-based wetland restoration. Ecological Applications 2706(8): e2706. https://doi.org/10.1002/eap.2706

Ussery JG, Krannitz PG (1998) Control of Scot's broom (*Cytisus scoparius* (L.) Link.): the relative conservation merits of pulling versus cutting. Northwest Science 72(4): 268–273. https://hdl.handle.net/2376/1207

Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, Pergl J, Schaffner U, Sun Y, Pyšek P (2011) Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities and ecosystems. Ecology Letters 14(7): 702–708. https://doi.org/10.1111/j.1461-0248.2011.01628.x

Wang H, Liu T, Dong H, Zhao W, Liu X, Wang R, Xu W (2022) Changes in the composition of the soil seed bank of grassland after giant ragweed (*Ambrosia trifida* L.) invasion. Journal of Environmental Management 317: 115468. https://doi.org/10.1016/j.jenvman.2022.115468

Weidlich EW, Flórido FG, Sorrini TB, Brancalion PH (2020) Controlling invasive plant species in ecological restoration: A global review. Journal of Applied Ecology 57(9): 1806–1817. https://doi.org/10.1111/1365-2664.13656

Werle R, Mobli A, DeWerff RP, Arneson NJ (2023) Evaluation of foliar-applied post-emergence corn–soybean herbicides on giant ragweed and waterhemp control in Wisconsin. Agrosystems, Geosciences & Environment 6(1): e20338. https://doi.org/10.1002/agg2.20338

Xian X, Zhao H, Wang R, Huang H, Chen B, Zhang G, Liu W, Wan F (2023) Climate change has increased the global threats posed by three ragweeds (*Ambrosia* L.) in the Anthropocene. The Science of the Total Environment 859: 160252. https://doi.org/10.1016/j.scitotenv.2022.160252

Yannelli F, Koch C, Jeschke J, Kollmann J (2017) Limiting similarity and Darwin's naturalization hypothesis: Understanding the drivers of biotic resistance against invasive plant species. Oecologia 183(3): 775–784. https://doi.org/10.1007/s00442-016-3798-8

Yannelli FA, MacLaren C, Kollmann J (2020) Moving away from limiting similarity during restoration: Timing of arrival and native biomass are better proxies of invasion suppression in grassland communities. Frontiers in Ecology and Evolution 8: 238. https://doi.org/10.3389/fevo.2020.00238

Yin P, Li JD, Yin H, Sun B, Sun JN, Wang GJ, Yan XF (2010) Impacts of *Ambrosia trifida* invasion plant biodiversity. https://www.cabidigitallibrary.org/doi/full/10.5555/20103148618

Supplementary material 1

Supplementary information

Authors: Chaeho Byun

Data type: docx

Explanation note: **fig. S1.** Sown species: Photographs of native seeds used in this study. **fig. S2.** Experimental layout (blocks) in two sites.

Copyright notice: This dataset is made available under the Open Database License (http://opendatacommons.org/licenses/odbl/1.0/). The Open Database License (ODbL) is a license agreement intended to allow users to freely share, modify, and use this Dataset while maintaining this same freedom for others, provided that the original source and author(s) are credited.

Link: https://doi.org/10.3897/neobiota.94.123363.suppl1

Supplementary material 2

Diversity index raw datasheets

Authors: Chaeho Byun

Data type: xlsx

Copyright notice: This dataset is made available under the Open Database License (http://opendatacommons.org/licenses/odbl/1.0/). The Open Database License (ODbL) is a license agreement intended to allow users to freely share, modify, and use this Dataset while maintaining this same freedom for others, provided that the original source and author(s) are credited.

Link: https://doi.org/10.3897/neobiota.94.123363.suppl2